Reviews in Fisheries Science, 8(2): 125–139 (2000)

Effects of Sedimentation and Turbidity on Lotic Food Webs: A Concise Review for Natural Resource Managers

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Abstract: Sedimentation and turbidity are significant contributors to declines in populations of North American aquatic organisms. Impacts to lotic fauna may be expressed through pervasive alterations in local food chains beginning at the primary trophic level. Decreases in primary production are associated with increases in sedimentation and turbidity and produce negative cascading effects through depleted food availability to zooplankton, insects, freshwater mollusks, and fish. Direct effects at each trophic level are mortality, reduced physiological function, and avoidance; however, decreases in available food at trophic levels also result in depressed rates of growth, reproduction, and recruitment. Impacts of turbidity to aquatic organisms often seem inconsistent among watersheds and experiments, but this apparent difference is actually due to the lack of correlation between suspended sediment concentrations (mg/L) and units of measure (Nephelometric Turbidity Units, NTU). The use of NTU as a surrogate measurement of suspended sediment to predict biotic effects within watersheds is dubious. Similar NTU measurements from different watersheds may be correlated with different concentrations of suspended sediment. For monitoring the effects of turbidity within local watersheds, we recommend that the correlation between suspended sediment and NTUs be examined over a range of discharge recordings, and that this be used as a baseline to examine local effects. We recommend that riparian buffer strips and livestock fencing be used to reduce sediment input to streams.

KEY WORDS: sedimentation, turbidity, lotic, food web.

I. INTRODUCTION

Karr et al. (1986) estimated that 50% of streams and rivers in the U.S. are impaired by habitat degradation. At the same time, drastic declines have occurred in all major aquatic faunal groups. Of North American freshwa-

* The unit is supported jointly by the U. S. Geological Survey, Virginia Department of Game and Inland Fisheries, Wildlife Management Institute, and Virginia Polytechnic Institute and State University.

1064-1262/00/$.50
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ter species, approximately 35% of amphibians, 37% of fishes, 73% of mussels, and 65% of crayfish have been classified as vulnerable, imperiled, or endangered (Williams et al., 1993; Richter et al., 1997). The decline of freshwater species appears to continue. During the 1980s, the number of North American fish species considered by the American Fisheries Society to be of special concern, threatened, or endangered increased by 30% (Williams et al., 1989). Among the causes for habitat degradation, sedimentation and turbidity are identified as important contributors to declines in aquatic faunas (Richter et al., 1997). Sources of stream sedimentation and turbidity include agriculture, forestry, mining, road construction, and urban activities. Sediment input to individual streams can be very large. For instance, stream sediment loadings from four logged watersheds in Oregon after road construction were estimated at 12,400, 8900, 4600, and 89 tons/mi^2-yr^{-1} (Brown, 1972). The magnitude of sedimentation to U.S. streams and rivers is placed in perspective by this example, considering that forest practices only account for an estimated 6% of national sediment pollution (Firehock, 1991). From cropland alone, approximately 17 tons ha^{-1} yr^{-1} of soil are eroded in the U.S. from combined water and wind causes, and an estimated 60% of this tonnage is deposited in streams and rivers (Pimentel et al., 1995). Cost estimates for in-stream and off-stream damages from soil erosion range from \$2.1 to \$10.0 billion yr^{-1} (Clark, 1985; Pimentel et al., 1995). Estimates include costs associated with recreation, water storage, navigation, commercial fishing, and property damage, but do not include costs of biological impacts (Clark, 1985).

Excessive sedimentation has been estimated to occur in 46% of all streams and rivers in the U.S. and is considered the most important factor limiting fish habitat (Judy et al., 1984). Other researchers have identified sedimentation as the most detrimental aquatic pollutant (Ritchie, 1972; Lemly, 1982). In a national survey of fishery biologists, turbidity was considered to be the most detrimental water quality characteristic to aquatic organisms and was estimated to affect 34% of all U.S. streams (Judy et al., 1984). The Environmental Protection Agency (EPA) (1994) has identified siltation as the most important source of water quality degradation. Increased turbidity has been found to be the strongest descriptor of reduced invertebrate density and biomass (Wagener and LaPerriere, 1985). Because sedimentation and turbidity are important contributors to declines in riverine fauna, resource managers need to be able to evaluate and predict local effects, but that is difficult if they must first conduct a full literature review. Our objectives are to provide a concise summary of the effects of sedimentation and turbidity on lotic

food webs and to discuss mitigating procedures that can be initiated to curtail biological impacts.

II. DEFINITIONS AND MEASUREMENTS

Eroded material that enters streams may be classified as particulate or that which remains in suspension. Particulates are silt, sand, and gravel that eventually settle to the substrata with decreasing water velocity and turbulence, resulting in sedimentation. Consequently, sedimentation is the process whereby substrata are covered and interstitial spaces of the substrata are filled by deposited sediment. Heavily sedimented lotic habitats are easily recognized and are characterized by a fine layer of clay, silt, and sand. Sedimentation of substrata may be measured in several ways, including (1) the area of streambed covered, (2) depth of coverage, (3) the size classification of sediment covering the substrata, and (4) percentage of interstitial spaces filled (Waters, 1995).

Turbidity, on the other hand, is caused by particles and dissolved substances in water, including organic and inorganic particulate and suspended matter, and dissolved substances that contribute to the color of water. The organic component may include algae. Turbidity may be defined as the properties of water that cause light to be scattered and absorbed (American Public Health Association [APHA], 1992). The sources of abiotic suspended matter can be directly eroded material, or sediments that have settled to the substratum and become entrained during periods of high flow. Suspended material is usually smaller than 62 μm (Waters, 1995). As stream order (size) increases, biotic contributions to turbidity increase (Vannote et al., 1980). In this article, our discussion primarily focuses on the effects of erosion-based sources of turbidity.

When one reviews the literature related to sedimentation and turbidity, it is apparent that turbidity is often used as a surrogate measure for estimation of sediment loads in aquatic environments (Newcombe, 1994; Newcombe and Jensen, 1996). In this literature, the reader may determine two sources of confusion concerning the measurement and reporting of turbidity. First, there are different methods used to measure turbidity, and therefore different units of measure are reported. Second, suspended and particulate concentrations (mg/l) are not necessarily correlated to reported units of measure (Gippel, 1989; Telesnicki and Goldberg, 1995). Therefore, confusion generated by the reporting of turbidity units that are not transferable is compounded by a lack of correlation between those units of measure and suspended and particulate load. Turbidity is not always a suitable surrogate measure of

suspended and particulate concentrations in water. This is because turbidity varies with the physical and optical properties of particulate and suspended matter (Gippel, 1989). Turbidity of similar water samples can also vary due to instruments used and the standards used to calibrate instruments, that is, Formazin vs. marl (Telesnicki and Goldberg, 1995).

Three different standard units of measure have been traditionally reported for turbidity. The unit of measure that is frequently encountered in older articles is Jackson Turbidity Units (JTU), measured by a Jackson candle turbidimeter. It should be noted that the APHA (1992) no longer recommends the measurement of turbidity using this technique. More recently, turbidity is measured using a nephelometric turbidimeter that measures the attenuation of a beam of light through a water sample. In other words, these instruments measure the absorption and scatter properties of light when it passes through water. Turbidity, as measured by this type of turbidimeter, is reported in either Nephelometric Turbidity Units (NTU) or Formazin Turbidity Units (FTU), depending on whether Formazin is used to calibrate the instrument (Telesnicki and Goldberg, 1995). The use of FTU in reporting turbidity is rarely seen in the literature. For the measurement of sediment concentrations below 25 mg/l, it has been recommended that marl be used as a standard for calibrating turbidimeters, rather than Formazin (Telesnicki and Goldberg, 1995). The APHA (1992) currently recommends that NTU be used as the standard of measure for reporting turbidity.

Because JTU, NTU, and FTU are reported in the literature with an implicit assumption of equivalency that may not exist, and because these units of measure are not necessarily correlated with sediment concentrations, Newcombe (1994) has presented a method of reporting biological effects of suspended sediment based on sediment exposure duration and concentration. In this method, the combination of duration and concentration is measured in mg * h/L that provides a dose response estimate. From this measure, a Severity of Effects (SE) is created from the natural log of mg * h/L whose values range from 0 to >17 (Newcombe, 1994; Newcombe and Jensen, 1996). Lower values of the SE denote lower combinations of sediment event duration and concentration. Newcombe (1994) uses the SE to describe a continuum of dose responses for various aquatic organisms. For purposes of this review, we find the association of the SE with detectable effects as somewhat impractical. For clarity, we will report general effects of sedimentation and turbidity on lotic food chains, and, where appropriate, supplement findings with turbidity in reported NTU and concentrations in mg/L. Again, the reader should regard recorded NTU with caution because values from different

streams and rivers may be correlated with different suspended concentrations of sediment. We recommend that NTU measurements be correlated to suspended sediment concentrations on an individual watershed basis.

III. EFFECTS

The effects of sedimentation and inorganic turbidity on aquatic habitat and biota are dependent on measured levels and persistence of sediment load (Cairns, 1990). High and sustained levels of sediment may cause permanent alterations in community structure, diversity, density, biomass, growth, and rates of reproduction and mortality. Impacts on aquatic individuals, populations, and communities are expressed through alterations in local food webs and habitat. The influence of increased sedimentation on lotic food webs is pervasive and begins at primary trophic levels.

Sediment in transport can have an abrasive quality and can reduce the quantity of periphyton that grows on stream substrata (Steinman and McIntire, 1990). Increases in lotic turbidity can limit light penetration and therefore reduce phytoplankton production (Hoetzel and Croome, 1994). Decreases in zooplankton can occur with very low levels of suspended sediment. At 6 NTU, McCabe and O'Brien (1983) found that feeding efficiency of *Daphnia pulex* was decreased by about 25%. Reductions in macrophyte biomass, growth, and diversity also have been shown to occur with higher levels of sediment and turbidity (Lloyd et al., 1987). As turbidity increases, a reduction in plant growth, including algae, occurs because less light is available for photosynthetic production (Kirk, 1985; Ryan, 1991). The effects of reduced primary production on herbivorous insects and fishes at higher trophic levels are compounded when sediment settles on remaining macrophytes. Thus, not only is primary production reduced by sedimentation and turbidity, but macrophyte quality also is reduced as a food source (Ryan, 1991). These impacts can occur with small increases in inorganic-based turbidity. Lloyd et al. (1987) found that an increase in turbidity of only 5 NTU decreased primary production by 3 to 13%, and increases of 25 NTU decreased primary production up to 50%. Not only does turbidity decrease available food sources for herbivores, but a reduction in phytoplankton also translates to a reduction in zooplankton (Lloyd et al., 1987). Consequently, a reduction in phytoplankton, and therefore zooplankton, may create cascading effects at higher trophic levels via a reduction in available food energy.

The influence of particulate and suspended load on aquatic insects is well documented. Macroinvertebrate density and diversity are directly

related to substrate diversity (Gore, 1985). As sediment settles, interstitial spaces between coarse substrata are filled, which reduces available habitat for macroinvertebrates (Lenat et al., 1981). If enough sediment fills interstitial spaces, then an impermeable sediment barrier may form that diminishes available living habitat and causes reductions in hyporheic oxygen levels due to the inhibition of interstitial water circulation (Beschta and Jackson, 1979; Gordon et al. 1992). Ryan (1991) concluded that a 12 to 17% increase in interstitial fine sediment may be associated with a 16 to 40% reduction in the total abundance of invertebrates. With sedimentation of the substrata, insect community structure may change with habitat alterations that favor burrowing insects tolerant of low oxygen levels (Ryan, 1991). Tsui and McCart (1981) found that densities and standing stocks of lotic insects were inversely related to levels of sedimentation. Wagener and LaPerriere (1985) reported that sedimentation decreased density and biomass in benthic macroinvertebrate communities, and stated that turbidity was the strongest descriptor related to such reductions. Insect escape through drift has been shown to increase with increases in substrata sedimentation (Rosenberg and Wiens, 1978; Culp et al., 1986). In addition, Rosenberg and Snow (1975) detected increased drift of macrobenthos with an increase in suspended sediment. With weekly exposure to sediment concentrations of 1700 mg/L, Fairchild et al. (1987) found changes in drift patterns and community structure of benthic insect communities and reported that recovery from this type of process exposure was gradual. Gammon (1970) found that shifts in benthic invertebrate communities were characterized by increases in silt-tolerant genera such as mayflies of Tricorythodes. These shifts were observed at suspended sediment concentrations as low as approximately 53 mg/L (Gammon, 1970).

While deposited and suspended sediments have been shown to negatively affect the survival of freshwater mussels (Unionidae), survival appears to be species specific. Ellis (1936), for example, showed that 0.6 to 2.5 cm of deposited silt resulted in significant mortality of yellow sandshell (*Lampsilis teres*), while threehorn wartyback (*Obliquaria reflexa*), mapleleaf (*Quadrula quadrula*), and monkeyface (*Q. metanevra*) were more resistant. The giant floater (*Pyganadon grandis*) had better survival than either Wabash pigtoe (*Fusconaia flava*) or black sandshell (*Ligumia recta*) when buried in detritus, sand, mucky sand, or silt (Imlay, 1972), and *F. flava* appeared to be more sensitive to silt and sand than either the plain pocketbook (*L. cardium*) or fatmucket (*L. siliquoidea*) (Marking and Bills, 1980). However, even *P. grandis*, a species often exposed to deposited sediment in lentic habitats, experienced high morality (85 to 100% mortality) when exposed to 45 cm of silt (Imlay, 1972). The eastern

elliptio (*Elliptio complanata*), a species of freshwater mussel commonly thought to be very tolerant of sedimented conditions, showed depressed growth rates in muddy substrates (Box and Mossa, 1998). It should be noted that many of the >30 extinct species of North American freshwater mussels resided in riffle and run habitats, including 15 species of the genus *Epioblasma* (Box and Mossa, 1998). It is probable that the decline of this genus is due to the loss and degradation of such habitats.

In an attempt to determine the mechanism by which sediment exposure leads to mortality of mussels, Aldridge et al. (1987) showed that exposures of 600 to 750 mg sediment/L reduced clearance rates and nitrogen excretion and increased O:N ratios for the pimpleback (*Q. pustulosa*), southern pigtoe (*F. cerina*), and Mississippi pigtoe (*Pleurobema beadleanum*). Aldridge et al. (1987) hypothesized that the effect of increased sediment exposure on these species was starvation due to decreased filtration rates. However, most studies dealing with the effects of sediment on freshwater mussels only have addressed survival after exposure to sediment, and placed little emphasis on the cause of mortality.

The marine literature contains numerous studies on the effects of sedimentation on bivalves that can be used to draw inferences about effects on freshwater species. Bayne et al. (1993) showed that the blue mussel, *Mytilus edulis*, compensates for increased levels of suspended sediment by (1) increasing filtration rates, (2) increasing the proportion of filtered material that is rejected, and (3) increasing the selection efficiency for organic matter. The Eastern oyster, *Crassostrea virginica*, also compensates for increased sediment loads by sorting particles and selective feeding, such that pseudofecal material produced by these animals contains less energy, nitrogen, and carbon/mg than ingested food (Newell and Jordan, 1983). Selective feeding in sedimented conditions does not appear to be common to all species of bivalves. The surf clam (*Spisula solidissima*), Iceland scallop (*Chlamys islandica*), and Atlantic pearl-oyster (*Pinctada imbricata*) have bulk pseudofecal production with limited selective rejection of unwanted food particles (Vahl, 1980; Robinson et al., 1984; Ward and MacDonald, 1996). Consequently, dietary absorption efficiencies for these species tend to decrease during exposure to sediment. Bricelj and Malouf (1984) concluded that species-specific responses to sediment are likely adaptations to sediment levels in the local environment, such that species inhabiting turbid environments are better able to select between organic and inorganic particles. Because many of the endangered freshwater mussel species have evolved in fast flowing streams with historically low levels of suspended sediment, such species may not be able to actively select between organic

and inorganic particles in the water column. Therefore, even low levels of sediment may reduce feeding and, in turn, have sublethal effects on unionid growth and reproduction. For example, fine sediment may interfere with feeding rates and indirectly affect growth through a reduction of photosynthetic production (Box and Mossa, 1998).

The effects of sedimentation and turbidity on ubiquitous taxonomic groups such as crayfish and snails are surprisingly undocumented in the scientific literature. Crayfish are presumed to be environmentally tolerant of episodic perturbations to habitat quality, but no testing of sedimentation effects has been conducted. Similarly, snails have not been tested for sediment tolerance, although effects on their food resources, such as benthic algae, undoubtedly affect gastropod populations. As dominant grazers in streams, any factors that reduce the abundance and availability of algae, or the suitability of substrata for feeding and mobility, will be detrimental to gastropod populations.

Sedimentation and turbidity can also contribute to decreases in local fish populations. Because fish can readily disperse, many species may simply relocate when sediment load is increased (Barton, 1977). For instance, avoidance of turbid waters has been observed in juvenile coho salmon (*Oncorhynchus kisutch*), arctic grayling (*Thymallus arcticus*), and rainbow trout (*O. mykiss*) (Newcombe, 1994; Newcombe and Jensen, 1996). For species that remain in the disturbed area, elevated levels of sediment may have an adverse effect on fish health. Increased sedimentation and turbidity can reduce dissolved oxygen in the water column, and in extreme cases may cause a thickening of the gill epithelium and reduced respiratory function (Horkel and Pearson, 1976; Goldes et al., 1988; Waters, 1995). Increased mortality has been associated with increased suspended sediment for arctic grayling (*T. arcticus*), Atlantic silverside (*Menidia menidia*), rainbow trout (*O. mykiss*), fourspine stickleback (*Apeltes quadracus*), white perch (*Morone americana*), yellow perch (*Perca flavescens*), American shad (*Alosa sapidissima*), striped bass (*M. saxatilis*), coho salmon (*O. kisutch*), chinook salmon (*O. tshawytscha*), chum salmon (*O. keta*), striped killifish (*Fundulus majalis*), sheepshead minnow (*Cyprinodon variegatus*), and even carp (*Cyprinus carpio*) (Newcombe, 1994; Newcombe and Jensen, 1996). As sedimentation occurs, spawning habitat also may be smothered. This is especially true for substrate spawners (Muncy et al., 1979). If sedimentation occurs after spawning, then oxygen supply to eggs and sac fry in the substrata may be decreased due to reductions in water circulation (Waters, 1995; Argent and Flebbe, 1999). Consequently, sedimentation decreases available spawning habitat, reduces spawning activity, and increases egg and larvae mortality (Alabaster and Lloyd, 1982; Ryan,

1991). An elevation in sac fry mortality of arctic grayling (*T. arcticus*) also has been shown to occur with increases in turbidity (LaPerriere, 1988; Reynolds et al., 1989). Reproductive strategies that involve parental care, such as fin fanning and egg nipping and mouthing, appear to be more successful in sedimented habitats, than those species that rely on substrate or pelagic broadcasting of eggs (Berkman and Rabeni, 1987). However, it should be emphasized that reproduction of warmwater species, such as largemouth bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*), and redear sunfish (*Lepomis microlophus*), can also be inhibited by higher levels of suspended sediment (Buck, 1956).

The feeding success of fish species that rely on visual search strategies can be affected in sedimented and turbid waters. In general, the abundances of visual feeders, like sunfish and trout, decline with elevations in turbidity (Gardner, 1981; Berkman and Rabeni, 1987). Gardener (1981) found that the feeding rate of bluegills (*L. macrochirus*) decreased at a turbidity of 60 NTU, and Breitburg (1988) noted striped bass larvae consumed less *D. pulex* with increased sediment concentrations. Species that depend on drifting insects as a major food source, including trout and salmon, have also shown depressed feeding rates (Ryan, 1991).

In summary, turbidity and sedimentation can have profound influences on the local ecology of lotic systems at the individual, population, and community levels. In such an environment, reductions in food availability, environmental quality, and habitat can directly affect growth, recruitment, and mortality rates at multiple trophic levels. In sedimented and turbid environments, reductions in species density, biomass, and diversity throughout a trophic level are translated into reductions in energy input to the next trophic level. Decreases in plant, zooplankton, and insect abundance and biomass initiate reductions in herbivore, omnivore, and, consequently, predator classes of fish (Berkman and Rabeni, 1987). Biotic adaptations to sediment perturbation also may lead to changes in local community composition. A further synergistic effect may occur when lotic communities are impacted by pesticides and other toxins entering the river or stream with eroded material (Lowrance et al., 1985).

IV. ***CONTROLS***

Many sediment control techniques have been used to reduce erosion and limit sediment input to streams and rivers. Some of the more prevalent methods include the implementation of fabric barriers, sediment traps and basins, water diversions, plantings, and proper road construction and maintenance (Waters, 1995). Although forestry, mining, roading, and construction activities are important sources of sediment to lotic environ-

ments, they are overshadowed by sediment input from agricultural sources (Waters, 1995). Because of this, we emphasize the importance of riparian buffer strips and livestock fencing to reduce sediment input.

Simply defined, riparian areas are vegetated corridors along rivers and streams. They may be considered important ecosystems. Lowrance et al. (1985) stated that "riparian ecosystems are the complex assemblage of organisms and their environment existing adjacent to and near flowing water. Riparian ecosystems are also a special class of wetlands." Riparian zones are often viewed as prime agricultural areas, both for crop and livestock production, because of seasonal nutrient enrichment by flooding (Lowrance et al., 1985). A major effect of riparian vegetation is the retardation of erosion by decreased surface water velocity that allows deposition of eroded material in the riparian zone before it enters the lotic environment (Lowrance et al., 1985; Schwab et al., 1993). In addition to sediment entrapment, riparian zones also filter nutrients from run-off for storage in plant material. They also provide bank stabilization and in-stream temperature regulation through shading. Because of their soil characteristics, riparian zones store large volumes of water. This water is released in a more even manner than in cleared riparian areas. Thus, lush riparian areas can facilitate consistency in annual flow patterns (Lowrance et al., 1985). Levels of suspended sediment increase quickly during storm events when riparian vegetation is absent (Schlosser and Karr, 1981). Whitworth and Martin (1990) compared streams with and without riparian filter strips and found that most stream sites with filter strips had a higher total number and taxa richness of macroinvertebrates. They also stated that sites with riparian strips also had higher species richness, diversity, total density, and index of biotic integrity (IBI) of fish (Whitworth and Martin, 1990).

Recommendations for optimal widths of streamside riparian zones vary in the literature. Published requisite widths for buffer strips are dependent on watershed use and hillside slope. Erman and Mahoney (1983) found that riparian buffer strips of <30 m were inadequate to protect streams from the effects of logging in Northern California mountain watersheds. To ensure proper function, Waters (1995) recommends a guideline width for riparian zones of 50 to 300 m, depending on local conditions. Although there are no hard and fast rules for the determination of requisite riparian strip widths, it is prudent to take a conservative management approach to ensure the link between riparian quality and stream biota. This is particularly true for near-stream flood plains. Wilkin and Hebel (1982) found that the majority of eroded material in a watershed came from cropped flood plains, more than from cropped uplands within watersheds. From this, it is

prudent to conclude that cropped land should not extent to the water's edge, and that riparian zones not be made available for grazing. We do emphasize that there is a trade-off between in-stream benefits and economic loss concerning the determination of riparian widths. Riparian widths that are larger than needed to inhibit the transmission of eroded material to aquatic environments remove valuable land from production. The area of research concerning the optimal width of riparian zones obviously needs further development. Where possible, we recommend that riparian zones, whatever width, be removed from crop and cattle production through fencing and the development of alternative water sources.

In the American Fisheries Society position statement on the effects of livestock grazing on riparian and stream ecosystems, overgrazing was listed as a significant source of degradation to riparian areas (Armour et al., 1991). Degradation of riparian areas by livestock reduces the sediment filtering function of these areas. Further impacts are often stream bank collapse and erosion due to bank trampling. Elimination of livestock grazing in riparian areas has been shown to have a restorative effect on stream biota. These benefits include increases in allochthonous input, increases of the standing stock and biomass of fish, increases in food for fish, and increases in cover for fish (Armour et al., 1991). Also, decreases in stream temperatures, reductions of sediment on substrata, increases in vegetative cover, decreases in average stream width, increases in average depth, and increases in bank stability have been shown to occur (Armour et al., 1991). In other words, the restoration of riparian zones results in significant positive effects on biotic and abiotic conditions in lotic environments.

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